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Snag dynamics and forest structure in Afromontane forests in KwaZulu-Natal, South Africa: implications for the conservation of cavity-nesting avifauna

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Little is known about snags in South African indigenous forests. Because of their value to wildlife, we initiated this study to quantify snag characteristics and dynamics. We also investigated forest structure (woody component and snags) and forest use by avian species of conservation concern. We examined tree species composition, abundance and tree size in eight Afromontane mixed *Podocarpus* forests in KwaZulu-Natal, South Africa. In addition, we investigated the succession of snags at the level of individual trees. Large *Podocarpus* species were less abundant in forests of the Dargle district than forests in the Creighton district. Mean diameter at breast height (dbh) of trees in the forests in these two districts differed significantly. Emergent and upper canopy trees (trees >50% of estimated 100% canopy height) were smaller in the Dargle district than those in the Creighton district. Although frequency of smaller trees (dbh = 10–60cm) compared with larger trees was high in both districts, forests in the Creighton district had more large trees. Snags were rare in all forests, but were larger (dbh) in forests of the Creighton district. The availability of large snags in these forests has implications for the conservation of secondary cavity nesting bird species including the Endangered Cape Parrot, *Poicephalus robustus,* as they are necessary for reproduction and recruitment of these birds.

Introduction

Snags (standing dead trees) are important in the ecology of many forest plant and animal species including fungi, mosses, lichens, invertebrates, birds and mammals (Bruns 1960, Gysel 1961, Haapanen 1965, Thomas *et al.* 1979, McComb *et al.* 1986, Dobkin *et al.* 1995). In southern Africa there is pressure to utilise forests, particularly the removal of snags for fuelwood (Shackleton 1993, Du Plessis 1995). However, studies have shown that the removal of snags for fuelwood impacts negatively on the diversity of cavity-using vertebrates in particular (Shackleton 1993, Du Plessis 1995). Furthermore, little is known about the density or dynamics of snags in southern African forests particularly if they are to be utilised sustainably.

Most studies have concentrated on the dynamics of snags in temperate forests especially in North America (Lee *et al.* 1997, Lee 1998, Ganey 1999, Moorman *et al*. 1999) and Europe (Kirby *et al.* 1998). There have been few studies of snag dynamics in more tropical forests (Gibbs *et al*. 1993). Studies show that snags provide nest, foraging and roost sites for birds (Sedgwick and Knopf 1990, Newton 1994, Ganey 1999). In North American coniferous forests, most studies have found a significant correlation between snag density and abundance of avian primary and secondary cavity nesters (Runde and Capen 1987, Schreiber and De Calesta 1992, Ohmann *et al*. 1994). Reductions in snag abundance affect populations of those species dependent on cavities (Ohmann *et al*. 1994). In South Africa, little is documented about indigenous forest snags and their relationship to the presence of cavity nesting birds especially given the rarity of some avian species native to these forests.

Natural forests in South Africa are defined as closed-canopy plant communities comprising woody plants more than 5m tall (Midgley *et al*. 1997). Unlike northern temperate forests, the forests in South Africa described as Afromontane or mistbelt mixed *Podocarpus* forests form a naturally fragmented series of forests. Typically in the province of KwaZulu-Natal (KZN), these forests vary in size from a few large forests of >1 000ha (e.g. Karkloof) to numerous small patches of c. 1ha found in small valleys (Cooper 1985, Cooper and Swart 1992, Low and Rebelo 1996, Midgley *et al.* 1997). Afromontane forests usually occur from c. 700–1 200m asl, on south facing slopes with a mean annual rainfall of c. 700–1 500mm (Pooley 1994).

Forests cover approximately 0.25–0.8% of the land surface of South Africa (Cooper 1985, Rutherford and Westfall 1994, Low and Rebelo 1996, Castley and Kerley 1996, Castley 1997, Geldenhuys and Von dem Bussche 1997, Midgeley *et al.* 1997). In the 18th and 19th centuries, logging

occurred in South African forests to supply an increasing demand from pioneer European settlers (Fourcade 1889, King 1941, Moll 1972, McCracken 1987, Lawes and Eeley 2000). Recently (<20 years) forest conservation and commercial afforestation have increased, so logging of indigenous timber has been reduced (Donald and Theron 1983). However, selective logging of dead trees in South African forests has continued, as they are considered redundant (Cawe and McKenzie 1989a). The importance of snags within forests has been neglected and there is concern over the removal of dead and dying trees for timber (Du Plessis 1995, Wirminghaus *et al.* 2001a, 2001b). The exploitation of the forests has had a significant effect on the structure and composition of forests of the region, with forests closest to earliest European populated areas being most adversely affected (Scott-Shaw 1971, Cawe and McKenzie 1989a, Lawes and Eeley 2000, pers. obs.). In many forests today, large *Podocarpus* spp. are almost absent, where once they formed a significant component in the climax community of these forests (Cawe and McKenzie 1989b, pers. obs.). It has been suggested that because of large tree removal the forest canopy mean height is approximately 60% of that in the past (T Oatley pers. comm.). This significant alteration to forest composition and structure has likely affected the ecology of these forests. For example, removal of large canopy trees has increased susceptibility to wind damage and the forest interior to desiccation (Moll 1972, Moll 1981, Cawe and McKenzie 1989b).

Despite the small area of forest in South Africa, 41 bird species (approximately 6.5% of South Africa's total avifauna excluding predominantly marine species) are classified as true forest or forest specific species (Oatley 1989, Symes *et al.* 2002). Such species are described as unable to survive in non-forest habitats (Oatley 1989). Therefore, forests in the region are relatively high in avifauna species richness and warrant particular conservation attention. Approximately 29% of the forest specific bird species in the Afromontane mixed *Podocarpus* forests of KZN are reliant on dead trees as breeding sites (Oatley 1989). Snags are often removed for apparent sustainable use of the forests with little understanding of their use to breeding cavity-nesting birds in the forests (Cawe and McKenzie 1989a, pers. obs.).

The Cape Parrot, *Poicephalus robustus*, an endemic to the mistbelt mixed *Podocarpus* forests of eastern South Africa, has suffered severe population declines in the past thirty years or more (Oatley 1989, Wirminghaus 1997, Wirminghaus *et al*. 1999, 2000, 2001a). Illegal capture for the avicultural market, shooting as pests and possibly disease has reduced populations to an estimated 500–1 000 birds (Downs and Symes 1998, Wirminghaus *et al*. 1999, 2000, 2001a, Downs 2000). Habitat degradation is also a likely cause of population declines but has not yet been quantified. As Cape Parrots are food and nest site specialists, habitat loss has likely resulted in food and nest site shortages (Wirminghaus *et al*. 2001b, 2001c, 2002).

Because little is known about snag density, size and degree of decay in South African indigenous forests, we initiated this study to quantify snag presence, characteristics and investigate dynamics. We studied existing forest structure (tree species composition, abundance and tree size) in eight mistbelt mixed *Podocarpus* forests in KZN, South Africa. We determined the availability and characteristics of snags and investigated the succession of snags at the level of individual trees. We expected snags to be rare and that the successional stage of each would vary. We related forest structure (as represented by snags) to the presence of cavity nesting birds given the rarity of some of the avian species native to these forests. We discuss the results in terms of implications for forest conservation, particularly avian secondary cavity-nesters found there.

Methods

Study sites

Eight mistbelt mixed *Podocarpus* forests were selected from two districts in KZN, Creighton (n = 4) and Dargle (n = 4) (Figure 1, Table 1). Although at similar altitudes, the forests varied in size, longitude and ownership (Table 1). It was difficult to quantify previous human impact and intensity of human intervention. The degree to which variation among forests was due to human impact or to natural differences was not ascertainable. However, Gxalingele Forest was not commercially logged by pioneer settlers (King 1941). Forests in the Dargle district were more exploited than forests in Creighton district (Donald and Theron 1983), but are experiencing less anthropogenic disturbance than they did historically (pers. obs.).

Forest structure and snag presence

We studied existing forest structure in terms of tree species composition, abundance and tree size in these eight mistbelt mixed *Podocarpus* forests. This included the presence and abundance of snags. A snag was defined as a standing dead tree, with diameter at breast height (dbh) >16cm (circumference at breast height >50cm). It was expected that snags were rare and the probability of finding them low. Consequently we walked multiple random transects in each forest, from boundary to boundary during July–August 2001.

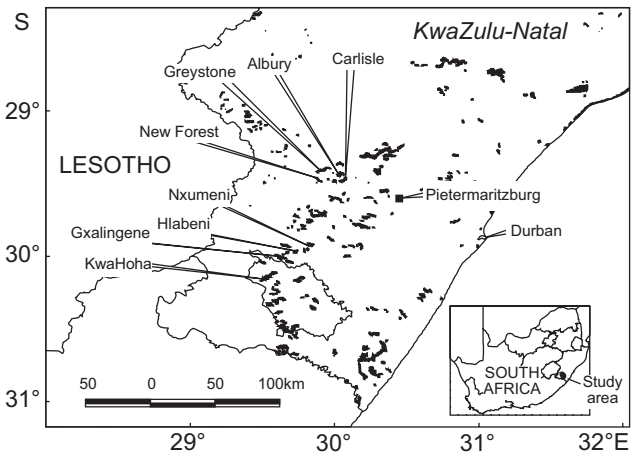

Figure 1: Distribution of forests investigated during this study. The distribution and presence of other forests in southern KZN is also shown

Table 1: Details of the forests in the study (After: Moll and Woods 1971, Cooper 1985, Cooper and Swart 1992, Department of Water Affairs and Forestry scanned 1: 50 000 maps); (SAFCOL — South African Forestry Company Limited; EKZNW — Ezemvelo KwaZulu-Natal Wildlife)

Forest	Area (ha)	Altitude (m asl)	Grid reference	Ownership
Creighton				
Hlabeni	258.4	1 140–1 700	29°58'20"S, 29°44'00"E	SAFCOL
Nxumeni	446.4	1 200–1 580	29°55'30"S, 29°50'20"E	SAFCOL
KwaHoha	314.2	1 280–1 720	30°09'40"S, 29°31'40"E	SAFCOL
Gxalingene	672.5	1 240–1 680	30°01'50"S, 29°37'40"E	EKZNW
Dargle				
Albury	129.3	1 300–1 500	29°26'30"S, 30°02'30"E	Private
Carlisle	155.5	1 220–1 380	29°28'00"S, 30°04'20"E	Private
New Forest	112.3	1 380–1 740	29°28'00"S, 29°52'50"E	Private
Greystone	192.5	1 460–1 720	29°24'40"S, 29°54'40"E	EKZNW

Transects were at intervals great enough to prevent overlap. More than five transects were walked for each forest. Along these transects, points at 100–150m intervals with a radius of 5m were surveyed. The distance between points varied because of the terrain of the areas. Although mainly used for bird censuses, point transects allow a patchy habitat to be sampled more easily (Buckland *et al*. 1994). Within each circular plot at each point, all trees with dbh >16cm or circumference at breast height >50cm were identified to species level. The dbh (cm) was measured and relative tree height (expressed as a percentage relative to average 100% canopy height) estimated. Snag characteristics (species, dbh and successional stage (Figure 2)) were recorded in each circular plot. Number of snags visible from beyond each circular plot area was also recorded as an additional index of snag abundance in the forest as snag presence in the circular plots was low.

Snag successional stages

We investigated the successional stages of snags at the level of individual *Podocarpus* trees. During January to March 1996, snags (n = 47), trees (n = 13) and partly dead trees (n = 6) were located and marked (so as to prevent re-recording later) in Hlabeni forest along random transects. The following data for each snag/tree were recorded: species, dbh and successional stage (Figure 2). In December 1997, an additional 24 living trees and 18 snags were measured, and changes in the previously recorded trees re-assessed (n = 7). Again in October 2001, previously marked trees and any changes in snag successional stage were assessed (n = 30).

Bird species

Avifaunal lists were generated for each forest using multiple techniques. These lists determined total number of forest species and number that nested in tree cavities (either primary or secondary). Presence of cavity nesting bird species was recorded while walking the forest transects, when visiting the forests on other occasions, and from ornithologists and bird watchers who had recorded bird species lists for the forests in the past 10 years.

Numbers of Cape Parrots at each forest were obtained from observations on the Cape Parrot Big Birding Day, an annual event initiated in 1997 to census remaining Cape Parrot populations (Downs and Symes 1998, Downs 2000, Downs unpublished data). Because censuses of Cape Parrots are difficult as they move between forest patches, a one-day national census was implemented to eliminate temporal variation, and cover all localities simultaneously. Groups of observers were assigned to specific forests. Two or more observers were positioned at vantage points to record presence or absence, arrival or departure, and roosting of Cape Parrots. Movement and activity recordings included number of Cape Parrots and direction of flight. Observations were made when Cape Parrots were most active, 3–5h after sunrise and 3h before sunset. When analysed, results were scrutinised to avoid any repeated counting of the same birds. As Cape Parrots are food nomadics, the maximum number of Cape Parrots seen at each locality in the morning and evening were determined.

Statistical analyses

Bray-Curtis Complete Link analysis was used to compare forests based on tree species abundance within each forest (McAleece *et al*. 1997). Other statistical analyses were completed using STATISTICA (Statsoft Inc. 1998). Data from each forest were combined so that n = 8.

Results

Forest structure and species composition

More than 45 tree species were identified in the forests (Table 2). Generally, the most abundant three tree species accounted for >50% of the trees (Hlabeni — 57.8%, Nxumeni — 55.2%, KwaHoha — 59.4%, Gxalingene — 68.3%, Albury — 50.1%, Carlisle — 48.0%, New Forest — 51.3%, Greystone — 39.5%).

Some tree species were common to most forests. These included *Xymalos monospora*, *Celtis africana* and *Eugenia zuluensis*, although these species may not have been abundant or present in all forests (Table 2). *Podocarpus* spp. were more abundant in the Creighton forests (Hlabeni, Nxumeni, KwaHoha and Gxalingene).

Forest similarity based on species abundance within each forest is shown in Figure 3. There was a distinct separation of Greystone and New Forest (Dargle district) from the other

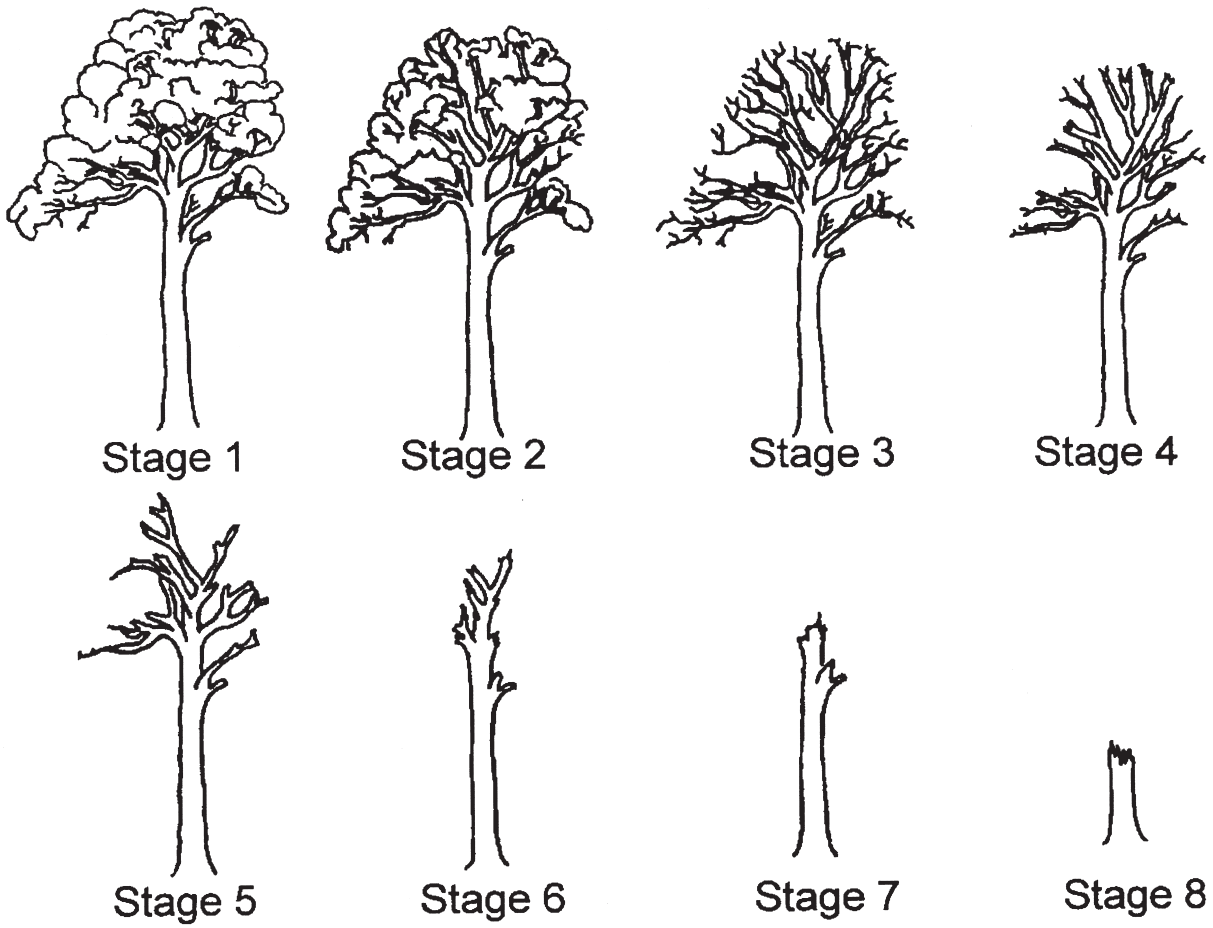

Figure 2: Successional stages of *P. falcatus* in forest (Downs and Symes unpubl. data) used in this study and applicable to most other forest tree species

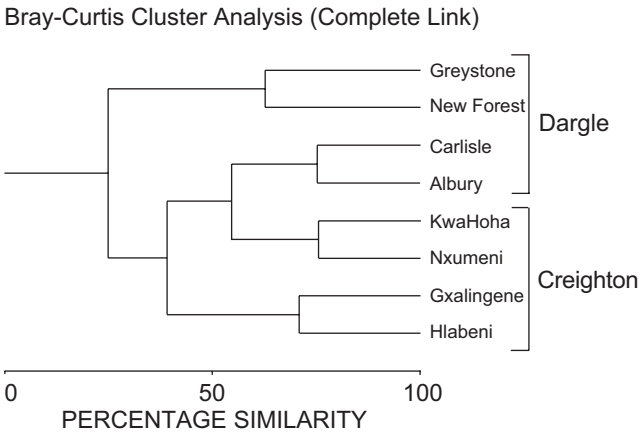

Figure 3: Dendrogram indicating forest similarity based on tree species abundance in each forest studied

forests. In the latter clump, Gxalingene and Hlabeni (Creighton district) separate from the remainder.

There was a significant difference in dbh of trees between forests (ANOVA, $F(1, 7) = 24.94$, $P < 0.05$, Table 3). A *post-hoc* Scheffe test indicated that most of the forests in the Creighton district, with the exception of KwaHoha Forest, had trees significantly greater in dbh than trees in the Dargle forests (Table 3). The differences in dbh size classes between forests possibly indicate differences in tree ages.

The highest number of trees occurred in the 20–29cm dbh size class in Hlabeni, KwaHoha, Gxalingele, Carlisle and Greystone forests (Figure 4). As trees of dbh <16cm were not recorded, a skewed distribution of tree size frequency within the forests is more likely. There was a lower frequency of trees in the higher dbh categories in the Dargle district forests (Figure 4e–h) compared with those in the Creighton district (Figure 4 a–d). The Dargle forests (Figure 4e–h) had no trees with dbh >120cm, while frequency of trees with dbh >120cm in the Creighton forests comprised 3.7% of all trees sampled.

There was a significant difference in the relative height size classes of trees between the forests (ANOVA, $F(1, 7) = 8.72$, $P < 0.05$, Table 3). The distribution of the relative height classes of trees for each of the forests is shown in Figure 5. In all forests there were greater number of trees in the 80–100% relative height class compared with the other height classes. In the Dargle no trees were classified as emergent (110–120%).

There was a significant difference in the dbh of trees in

Table 2: Relative species abundance in the study forests (number of trees recorded during sampling as a fraction of the total number of trees recorded while sampling). The three most abundant species for each forest are indicated in bold. The forests are: 1 = Hlabeni, 2 = Nxumeni, 3 = KwaHoha, 4 = Gxalingene, 5 = Albury, 6 = Carlisle, 7 = New Forest, 8 = Greystone. See Figure 1 for forest locations

Species	Creighton District				Dargle District			
	1	2	3	4	5	6	7	8
Apodytes dimidiata	–	–	–	–	–	0.8	–	–
Buddleia salvifolia	–	–	0.8	–	–	–	1.9	0.7
Calodendrum capense	2.5	4.8	–	–	7.3	4.7	–	4.2
Calpurnea aurea	–	–	–	–	–	–	–	0.7
Canthium ciliatum	–	–	–	–	–	0.8	–	0.7
Carissa bispinosa	0.6	–	–	–	–	–	–	–
Cassine pappilosa	–	–	–	–	1.2	–	–	–
Celtis africana	9.4	**11.4**	**12.8**	6.8	**9.8**	**14.2**	5.1	4.2
Clausena anisata	–	–	–	–	–	–	0.6	1.4
Stropanthus speciosus	–	1.0	0.8	–	–	1.6	–	2.1
Combretum kraussii	–	–	0.8	–	4.9	0.8	–	2.1
Cryptocarya woodii	–	–	–	0.7	–	1.6	3.2	4.2
Cussonia sphaerocephala	0.6	–	0.8	0.7	–	–	–	–
Dais cotinifolia	–	–	–	–	–	–	–	2.8
Diospyros whyteana	0.6	–	–	1.4	–	1.6	3.8	4.9
Dombeya tiliacea	–	–	–	–	–	–	0.6	0.7
Ekebergia capensis	–	–	0.8	–	–	–	–	–
Erythroxylum pictum	–	–	–	–	–	–	0.6	–
Euclea crispa	–	–	–	–	–	–	–	2.8
Euclea natalensis	–	–	–	–	–	–	–	0.7
Eugenia zuluensis	**27.0**	3.8	6.0	**25.7**	4.9	4.7	**14.1**	**12.0**
Ficus craterostoma	–	–	–	0.7	–	–	–	–
Gymosporia mossambicensis	3.1	1.9	–	0.7	4.9	8.7	0.6	3.5
Halleria lucida	1.3	–	3.8	1.4	6.1	4.7	1.3	1.4
Heteromorpha trifoliata	–	–	–	–	1.2	–	–	–
Kiggelaria africana	6.3	9.5	4.5	2.7	**9.8**	**9.4**	5.1	**10.6**
Ocotea bullata	–	–	0.8	–	–	–	–	–
Olea capensis	–	–	–	–	–	–	5.1	–
Podocarpus falcatus	**13.2**	4.8	2.3	8.1	1.2	5.5	1.3	–
Podocarpus henkelii	3.1	**21.9**	**30.1**	**28.4**	–	0.8	–	–
Podocarpus latifolius	2.5	–	0.8	1.4	2.4	1.6	**17.3**	–
Prunus africana	0.6	–	–	–	1.2	1.6	–	1.4
Ptaeroxylon obliquum	3.1	3.8	6.0	3.4	**9.8**	4.7	**19.9**	**16.9**
Rapanea melanophloeos	0.6	–	–	–	–	–	–	2.8
Rhus spp.	–	–	–	–	–	–	0.6	0.7
Rothmania capensis	0.6	–	–	–	–	–	–	–
Scolopia mundii	0.6	1.9	2.3	–	2.4	0.8	10.3	8.5
Scolopia zeyheri	–	–	–	–	–	0.8	–	1.4
Scutia myrtina	–	–	–	–	–	–	3.2	–
Trimeria grandifolia	–	–	–	–	1.2	–	0.6	2.8
Vepris lanceolata	1.3	1.0	0.8	2.7	1.2	–	–	–
Xymalos monospora	**17.6**	**21.9**	**16.5**	**14.2**	**20.7**	**24.4**	–	–
Zanthoxylon davyi	0.6	9.5	6.0	–	1.2	0.8	1.3	0.7
Pinus patula and *Sequoia* sp. *	–	1.0	1.5	–	–	–	–	–
Solanum mauritianum *	–	–	–	–	4.9	1.6	–	–
Unidentified	0.6	–	–	–	–	–	–	0.7
Snag	3.8	1.9	2.3	1.4	3.7	3.9	3.2	4.2

* alien species

each of the relative height classes for all forests combined (ANOVA, df = 11, F = 48.9, P < 0.05, Figure 6). In particular, the relative height classes 100% and 110% were significantly different to most other relative height classes (Scheffe test, Figure 6). This showed that canopy and emergent trees had a higher dbh.

Tree basal area density (m^2 ha^{-1}) and tree density (tree ha^{-1}) for each of the forests is shown in Table 3. Despite New Forest and Greystone forests having trees with the lowest dbh, they had the highest tree density. There was no difference in tree density (trees ha^{-1}) between forests in the two districts (t-test indep. samples, t = –0.59, df = 6, P > 0.05), but there was a significant difference in basal area density between the forests in the two districts (t-test indep. samples, t = 4.55, df = 6, P < 0.05, Table 3). Mean (± SE) basal area of forests in the Creighton district was 88.95 ± 10.27m^2 ha^{-1}

Table 3: The number of quadrats sampled per forest, dbh of trees, basal area of trees within each forest and density of trees in respective forests (sample sizes of trees in parentheses)

Forest		Quadrats sampled	Tree mean dbh (cm ± se)	Tree basal area density (m^2 ha^{-1})	Tree density (tree ha^{-1})
Hlabeni	(n = 159)	54	51.4 ± 2.7	112.2	374.9
Nxumeni	(n = 105)	42	53.5 ± 2.9	93.1	318.3
KwaHoha	(n = 133)	44	38.9 ± 2.0	62.3	384.9
Gxalingene	(n = 148)	73	56.1 ± 2.9	88.2	258.1
Albury	(n = 82)	44	36.3 ± 2.1	31.1	237.3
Carlisle	(n = 127)	45	37.2 ± 1.8	50.6	359.3
New Forest	(n = 156)	44	28.9 ± 1.0	35.1	451.4
Greystone	(n = 142)	43	30.2 ± 1.2	36.6	420.4

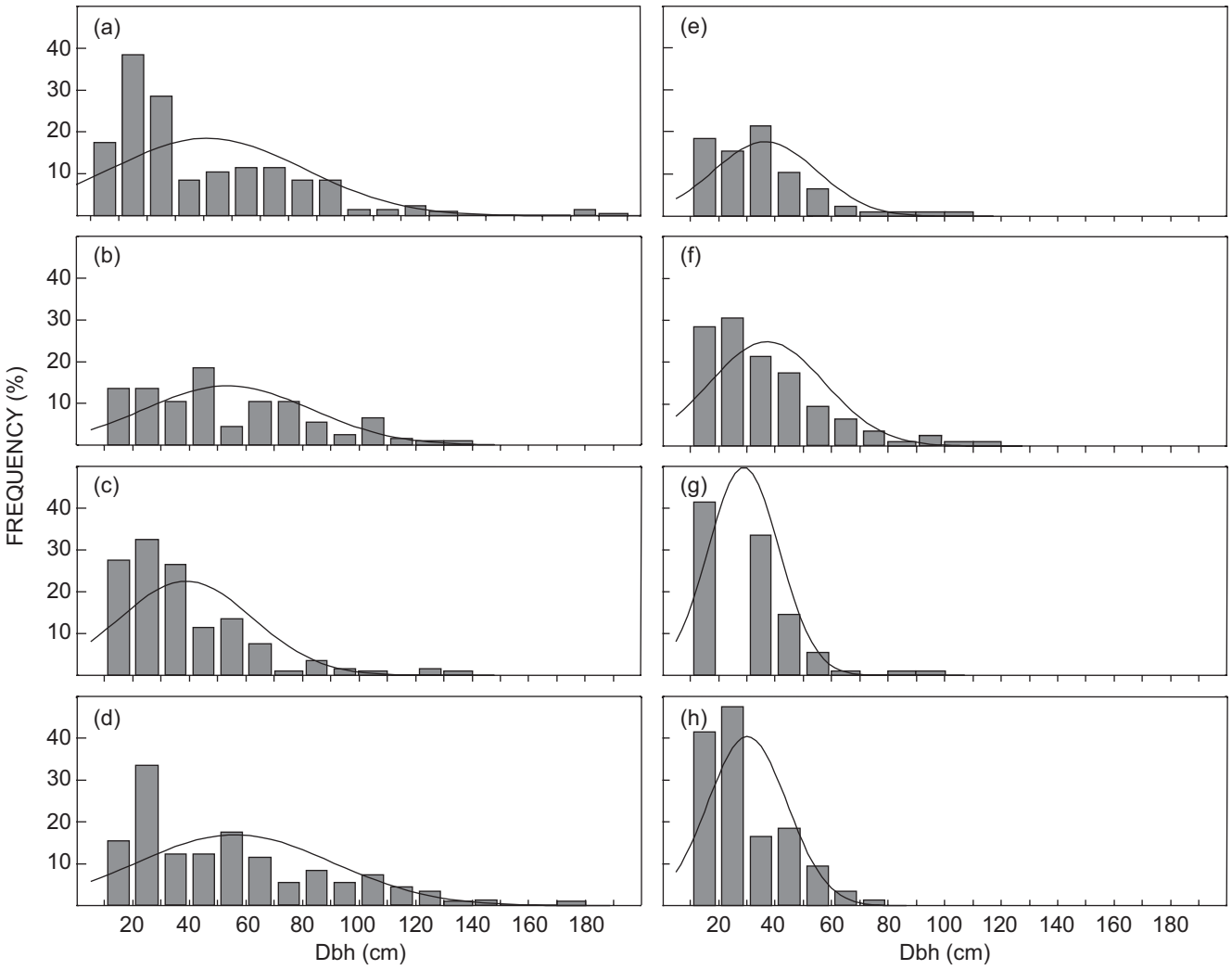

Figure 4: Frequency of dbh tree classes in each of the forests where: a = Hlabeni, b = Nxumeni, c = KwaHoha, d = Gxalingene, e = Albury, f = Carlisle, g = New Forest, h = Greystone

(n = 4) compared with the forests in the Dargle district of 38.34 ± 4.25m^2 ha^{-1} (n = 4).

Forest snags

The forests showed a range in the number of snags as a percentage of trees sampled, in the percentage of quadrats in which snags were found and in the percentage of quadrats from which snags were visible (Table 4). However, snags were rare in all forests (<4.2%) of the trees sampled. There was a significant difference in snag dbh between forests (ANOVA, $F(1, 7) = 7.58$, $P < 0.05$). Most snags had

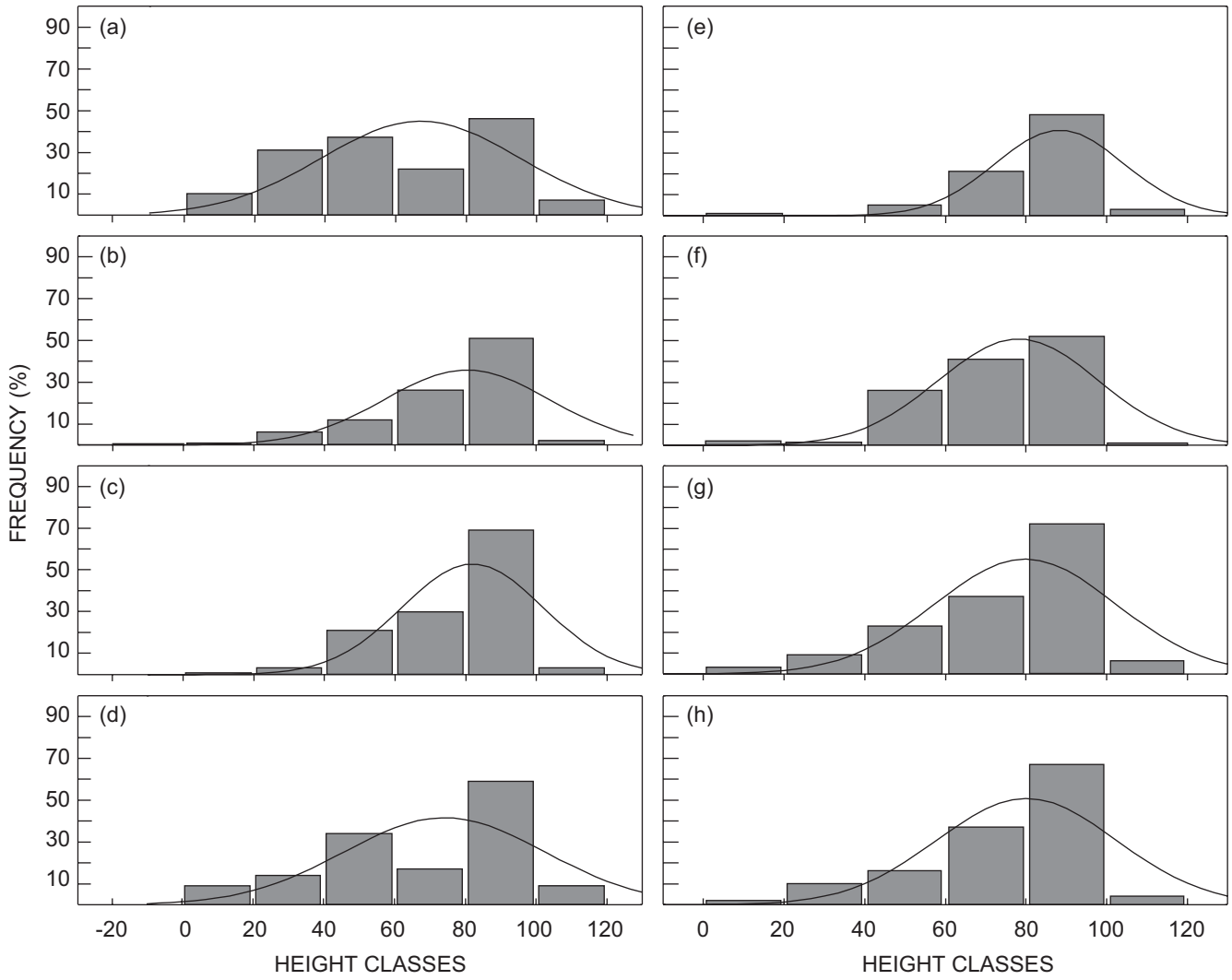

Figure 5: Frequency of relative tree height classes in each of the forests where: a = Hlabeni, b = Nxumeni, c = KwaHoha, d = Gxalingene, e = Albury, f = Carlisle, g = New Forest, h = Greystone

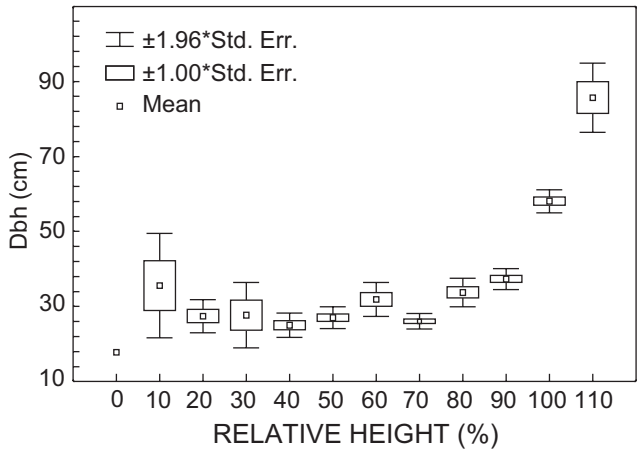

Figure 6: Dbh of trees versus the relative height classes for all the forests studied

a small dbh of 20–40cm (Figure 7a) and most were relatively short (Figure 7b). Larger snags were found in Gxalingene forest (Table 4). These snags usually had nest holes suitable for Cape Parrots or hornbills. During the present study of the forests, only two snags were found that had nesting Cape Parrots. At Hlabeni forest, one of the marked snags (stage 6, Figure 2) was used for the first time by breeding Cape Parrots in August 2001. The nest was monitored but it blew over in strong winds and rain during October 2001 killing the chicks before they fledged.

Snag succession

Marked snags measured at Hlabeni forest to monitor snag succession had a mean dbh = 77.6 ± 4.8cm (n = 38) which was greater than that measured in the quadrats in this forest. This is possibly an artefact of the sampling method where smaller snags were less likely to be detected when walking through the forest and so were not marked and monitored over time.

Table 4: Characteristics of snags in each forest showing number of snags recorded at each forest as a percentage of trees sampled, percentage of quadrats in which snags were recorded, percentage of quadrats from which snags were visible (number recorded in parentheses) and snag mean dbh (± se) (sample size in parentheses)

Forest	Snags as a percentage of trees sampled	Percentage of quadrats in which snags were recorded	Percentage of quadrats from which snags were visible	Snag mean dbh (cm ± se)
Hlabeni	3.8	11.1	31.5 (17)	52.5 ± 12.6 (6)
Nxumeni	1.9	4.8	11.9 (5)	43.1 ± 11.0 (2)
KwaHoha	2.3	6.8	6.8 (3)	36.0 ± 11.4 (3)
Gxalingene	1.4	2.7	16.4 (12)	125.3 ± 1.1 (2)
Albury	3.7	6.8	4.5 (2)	30.7 ± 6.1 (3)
Carlisle	3.9	11.1	11.1 (5)	29.5 ± 8.7 (5)
New Forest	3.2	11.4	4.5 (2)	33.4 ± 6.2 (5)
Greystone	4.2	14.0	0	22.2 ± 1.2 (6)

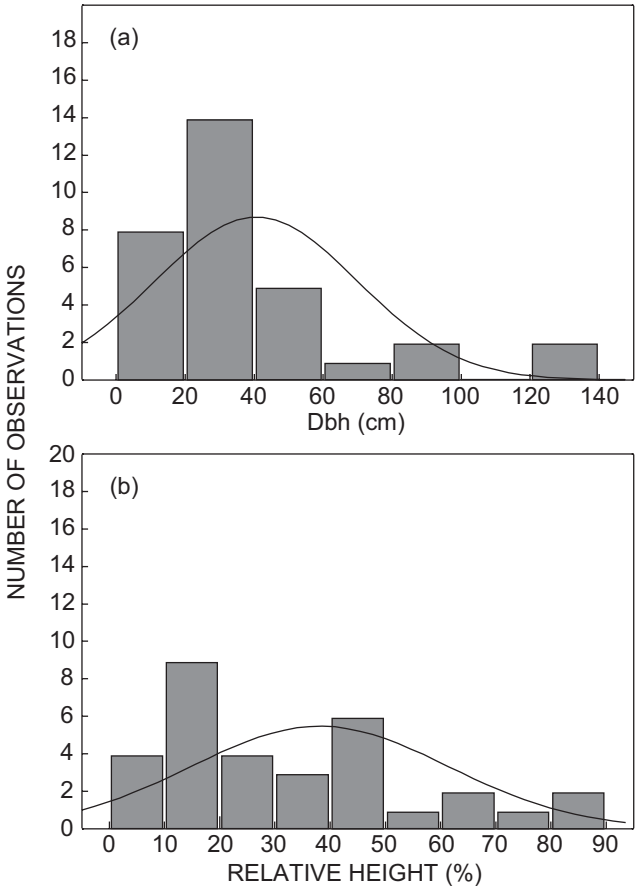

Figure 7: (a) Frequency of dbh classes of snags in all of the forests; (b) Frequency of relative snag tree height classes in all of the forests

Succession of *Podocarpus* snags generally occurred in a predictable sequence (Figure 2). Death, caused by heart-rot or lightning was followed by leaf loss (Figure 2, stage 2), bark loss (Figure 2, stages 3–6) and gradual loss of branches due to wind and decay. Snags appear to be susceptible to being blown over completely during stages 4–7, hence the reduction in number of snags, especially in stages 7–8.

Frequency of snags in Hlabeni forest at each successional stage was expected to decrease from stage 1 to stage 8. However, the greatest frequency of snags was at stages 4–7 (Figure 8a). The rates that snags progressed to the various successional stages varied as shown by Figure 8b.

Forest cavity nesting avifauna

During the study, the Cape Parrot, as well as other large cavity nesting species, were seldom seen in the Dargle forests. Sampling techniques used in all forests, did not allow detailed analysis especially for smaller cavity nesting species. It is, however, evident that larger cavity nesting species, including the Cape Parrot and hornbill species, are less abundant in the forests of the Dargle district compared with the other forests studied (Table 5).

Discussion

Forest snags and cavity nesting avifauna

Snags appear to be more common in North American temperate forests than the forests studied here (Lee *et al.* 1997, Lee 1998, Ganey 1999, Moorman *et al.* 1999). Snags were generally scarce in the present study. In North American studies, it has also been found that large snags (≥38cm dbh and ≥23m tall) are used more by cavity nesting birds than smaller snags (Scott 1978). In the present study, few snags were suitable for cavity nesters such as the Cape Parrot and hornbill species, which generally require a large cavity in a tall snag (Wirminghaus *et al.* 2001b, pers. obs.). Larger snags were found in Gxalingene forest where the greatest numbers of Cape Parrots have been recorded regularly roosting and nesting (Downs unpublished data). Cape Parrots have been recorded breeding at Hlabeni forest in snags of successional stages 4–7 (Wirminghaus *et al.* 2001b, Downs and Symes unpubl. data). These snag successional stages appear to be those most likely to be used by breeding Cape Parrots.

Mature trees that were logged as a source of timber in the past are trees that would now be dying, providing cavities and refuges for snag reliant species. It is possible that trees that are now providing these sites are either ‘second grade’ trees not selected by loggers in the past, or were in protected or inaccessible areas. It is also presumed that trees are not attaining ages that they once did in the past. With a reduced canopy height, emerging trees are more suscepti-

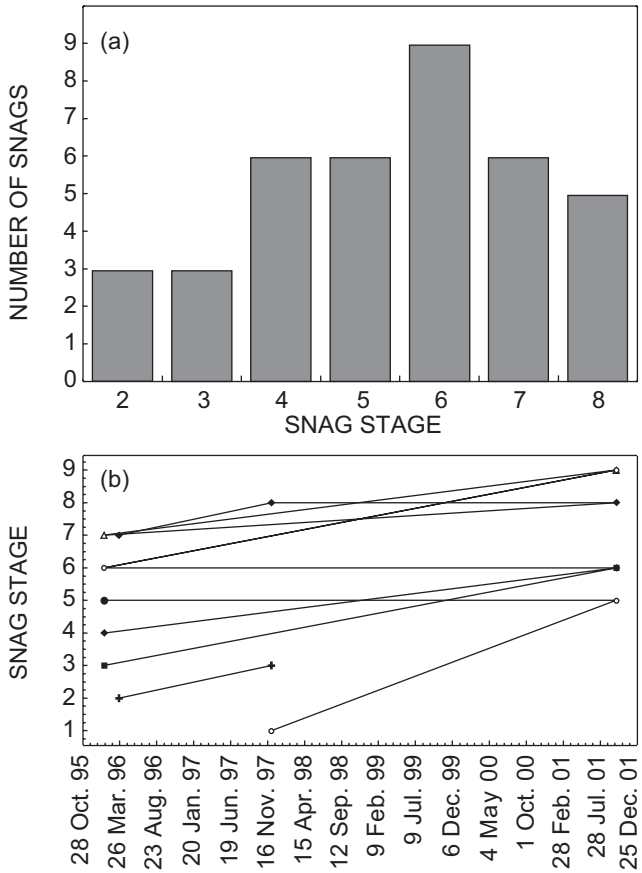

Figure 8: (a) Number of snags at each successional stage sampled in Hlabeni forest (22 January–24 March 1996); (b) Snag succession as shown by changes in different sampled *P. falcatus* snag stage over time at Hlabeni forest

ble to wind damage before reaching maximum age and height, and hence unable to reach maximum longevity. This would in turn result in a decrease in the size and quality of snags that are available to certain species, particularly the Cape Parrot, for breeding.

Snag succession has not been documented in forests of South Africa. It is likely that a snag will provide an optimal nesting tree for a cavity nester for only a short period. The survival of a snag (predominantly *Podocarpus* spp.) from stages 4–7 (stages suitable for nesting) is estimated at c. 6.5 years. In North American studies of snags, those in later decay classes were dominant (Ganey 1999). However, these were also more susceptible to damage, and were then less suitable for birds as nest sites (Ganey 1999).

Tropical forests have fewer snags compared with temperate forests but have more cavity-nesting species (Gibbs *et al*. 1993). With low snag availability, sites for cavity-nesters in forests are expected to be few, so there is increasing competition for optimum cavity nest sites. If nest sites are limiting, then recruitment is expected to be low. The forests of this study had six species that were cavity nesters. Only one of these, the Olive Woodpecker (Table 5), is an excavator. Consequently snags form an important component in these forests for Cape Parrot and the other cavity nesting species (Table 5).

The importance of snags (and *Podocarpus* spp.) for the survival of the Cape Parrot has been highlighted (Wirminghaus *et al.* 1999, 2000, 2001c, 2001d). Snags suitable for breeding Cape Parrots appear to be rare in these forests (pers. obs.). It is therefore important that they are not removed from forests (e.g. timber or fire wood) and are retained as an integral component in the forest ecology. Since 1993 only three Cape Parrot nest sites, all in dead *P. falcatus*, have been located in Hlabeni forest. In each a maximum of three, two and one breeding attempts were recorded, with the latter attempt unsuccessful (Wirminghaus *et al.* 2001b, Downs and Symes unpubl. data).

The responses of forest avifauna to disturbance as a result of timber harvesting have been well documented (see Kutt 1996). Although densities of cavity nesters were not quantified in this study, preliminary presence/absence data and maximum numbers of particular species can be used to interpret forest quality for cavity nesters. The probability of birds in a forest increases with substrate available for cavity excavation i.e. snag basal area (Swallow *et al.* 1986). This in turn indicates that the probability of forest use by cavity nesters is higher for more mature forests (Swallow *et al.* 1986), as indicated in this study. However, forest characteristics are better predictors of cavity nest use than snag characteristics (Swallow *et al.* 1986). Therefore, forest condition can be used as a predictor of presence of cavity nesters (i.e. Cape Parrot), and appropriate conservation management plans initiated.

Forest tree composition

East African forests show marked differences in tree species composition between forests sites separated by <15km (Chapman *et al.* 1997). This was also shown in KZN forests where different forest type floristic compositions have been classified (MacDevette *et al.* 1988). MacDevette *et al.* (1988) grouped Nxumeni forest within the Dargle forests based on floristic composition, which was not found in this study (Figure 3). Insufficient forests were sampled to warrant any classification of forest types, yet, based on species abundance of trees within each forest, the floristic distinctiveness of each forest is clear. These findings are in accordance with earlier descriptions of forests where distinct contrasts in floristic composition and structure between forest patches were recognised (Sim 1907, White 1978). Despite this, it is likely that *Podocarpus* spp. were the dominant species present in most Afromontane forests of the region (Moll and Woods 1971, Donald and Theron 1983, Cawe and McKenzie 1989a). These large forest canopy emergent trees have been particularly impacted, evident by their absence in the Dargle forest complex. The rarity of tree species once abundant in these forests is likely to have important ecological consequences in the region. There was an absence of large trees in the forests at Dargle. This suggests, together with the number of young trees observed, that selective logging of trees has significantly altered mean tree size (dbh) within these forests. A high frequency of small (dbh) trees in the Dargle forests suggests recent large tree removal and subsequent increased regeneration rates.

Table 5: Cavity nesting species recorded in respective forests of this study (* indicates secondary cavity nester, [1] indicates large cavity nester and potential nest site competitor of Cape Parrot, [2] indicates forest specific species). Presence indicated by p, with maximum number certain species given (Oatley 1989, Maclean 1993). (Forests: 1 = Hlabeni, 2 = Nxumeni, 3 = KwaHoha, 4 = Gxalingene, 5 = Albury, 6 = Carlisle, 7 = New Forest, 8 = Greystone)

English name	Scientific name	Creighton forests				Dargle forests			
		1	2	3	4	5	6	7	8
PSITTACIDAE									
Cape Parrot*[2]	*Poicephalus robustus*	38	12	29	100	5	5	2	0
TYTONIDAE									
Barn Owl*[1]	*Tyto alba*	p							
STRIGIDAE									
Wood Owl*[1,2]	*Strix woodfordii*	p	p	p	p		p		p
TROGONIDAE									
Narina Trogon*[1,2]	*Apaloderma narina*	p	p	p	p	p	p		p
UPUPIDAE									
Hoopoe*	*Upupa epops*			p			p		
PHOENICULIDAE									
Redbilled Woodhoopoe*	*Phoeniculus purpureus*						p		
BUCEROTIDAE									
Trumpeter Hornbill*[1,2]	*Bycanistes bucinator*	3		1	p				
Crowned Hornbill*[1]	*Tockus alboterminatus*	3	p	2	1				
Southern Ground Hornbill*	*Bucorvus leadbeateri*	2		p	5				
LYBIIDAE									
Blackcollared Barbet	*Lybius torquatus*			p					
Redfronted Tinker Barbet	*Pogoniulus pusillus*	p		p	p			p	p
INDICATORIDAE									
Scalythroated Honeyguide*[2]	*Indicator variegatus*	p	p						
Sharpbilled Honeyguide*	*Prodotiscus regulus*	p							
PICIDAE									
Cardinal Woodpecker	*Dendropicos fuscescens*	p	p						
Olive Woodpecker[2]	*Mesopicos griseocephalus*	p	p	p	p	p	p	p	p
JYNGIDAE									
Redthroated Wryneck*	*Jynx ruficollis*							p	
PARIDAE									
Southern Black Tit*	*Parus niger*	p	p	p	p	p	p	p	p
STURNIDAE									
Glossy Starling*	*Lamprotornis nitens*	p							
Redwinged Starling*	*Onychognathus morio*	p	p	p	p		p	p	p

Predominant tree species removed from forests in KZN by pioneer European settlers included large canopy trees like *Podocarpus* spp., *Ptaeroxylon obliquum* and *Ocotea bullata* (Lawes and Eeley 2000). Very few forests were unaffected by logging, yet for logistical reasons forests closest to main trade routes were most affected. Also, forests closer to earliest settlements were likely impacted longer, resulting in greater ecological damage. Despite the earlier removal of *Podocarpus* spp. and *P. obliquum*, these species were present in the forests studied but the former were more abundant in the forests of the Creighton district while the latter was more abundant in the forests of the Dargle district. Low abundance of *Podocarpus* spp. in the Dargle forests (Albury, Carlisle, New Forest and Greystone, Table 2) possibly reflects selective removal of these tree species in the past, as reported historically.

Xymalos monospora ranks within the top three abundant species of most of the forests studied with the exception of New Forest and Greystone forests (Table 2). It has been suggested that this species becomes dominant in heavily disturbed areas (Palmer and Pitman 1972, Pooley 1994). Also, Moll and Woods (1971) suggest that seral stands of *X. monospora*, *Zanthoxylon davyii* and *Kiggelaria africana* precede climax stands of *P. henkelii* and *P. falcatus*. However, we recorded *X. monospora* in association with climax communities of *Podocarpus* spp. within the Creighton forests (forests 1–4, Table 2). Similarly, Cawe and McKenzie (1989b), recorded this species as dominant in relatively undisturbed *X. monospora–O. bullata* associations. In the present study, *O. bullata* was only recorded in one of the forests studied, KwaHoha forest. Presence of this species in forests has suffered greatly because of de-barking for the traditional medicine trade (Cooper 1985, Cawe and McKenzie 1989b). It is likely almost extinct in many forests where it was once abundant (Oatley 1984).

Forest structure

Evidence of many young trees suggests that selective logging of old trees has significantly altered mean tree size (dbh) within these forests. This in turn has affected basal area between complexes, indicating that forests most accessible to exploitation have been most affected. In the Transkei forests (Cawe and McKenzie 1989c), high basal area was

attributed to larger canopy trees as was found in the forests of the Creighton district in the present study. In addition, the estimation of basal area for the forests in the Creighton district was significantly larger than that identified for Afromontane forests (Huntley 1984), suggesting older trees and possibly a less disturbed forest regime here. Basal area within the Dargle forests was lower than that identified as Afromontane forest (southern Cape, 40–50m^2 ha^{-1}) (Huntley 1984), suggesting younger trees and possibly higher levels of large tree removal.

Forest strata

The absence of large trees and the higher presence of young trees further suggest a higher level of selective removal of canopy and emergent trees in the past from the Dargle district forests. There was little difference in understorey tree dbh between all forests in the present study. However, there was a higher frequency of smaller (dbh) trees in the Dargle forests suggesting regeneration processes of the canopy component.

Management implications

Previous authors have highlighted the importance of conserving forest for various reasons other than conserving cavity-nesting species (Cawe and McKenzie 1989a, Castley and Kerley 1996). Future studies will be required to determine more specific relationships between cavity nesting avifauna, snag characteristics and dynamics and their importance within the forest biome. Further detailed investigations into the snag composition of other forests may clarify snag processes and dynamics within Afromontane forests in the region. In particular, information on input-, decay- and fall-down-rates of snags, as well as cavity turnover and densities are required for South African forests.

There are likely very few mature pristine mistbelt mixed *Podocarpus* forests remaining today, with most being affected by tree removal in the past. In line with the principles of sustainable use, it is doubtful whether any commercial exploitation of the forests in this study (especially those of the Dargle district that were previously heavily exploited) would be feasible. With a recovery period of >60 years, regeneration may still not be complete. Stem densities are small and abundance of commercially viable species is low, especially in more utilised forests (viz. Dargle forests). Also, little is known about regeneration and growth rates (Moll and Woods 1971). In addition, as snags are rare, their removal for firewood or timber would be short term and have an impact on cavity nesting species. Understanding the long-term successional stages of forests in the region can assist in management and conservation strategies for the long-term survival of forest biodiversity.

Trial plots of *Podocarpus falcatus* have shown economic viability to equal or exceed that of exotic timber species (Geldenhuys and Von dem Bussche 1997). The development of indigenous plantations, especially in areas where forests have been cleared, will contribute to the overall economy of the region and the conservation and regeneration of indigenous forest.

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